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A future perspective on lithium-ion battery waste flows from electric vehicles

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ABSTRACT

As a proactive step towards understanding future waste management challenges, this paper presents a future oriented material flow analysis (MFA) used to estimate the volume of lithium-ion battery (LIB) wastes to be potentially generated in the United States due to electric vehicle (EV) deployment in the near and long term future. Because future adoption of LIB and EV technology is uncertain, a set of scenarios was developed to bound the parameters most influential to the MFA model and to forecast "low," "baseline," and "high" projections of future end-of-life battery outflows from years 2015 to 2040. These models were implemented using technology forecasts, technical literature, and bench-scale data characterizing battery material composition. Considering the range from the most conservative to most extreme estimates, a cumulative outflow between 0.33 million metric tons and 4 million metric tons of lithium-ion cells could be generated between 2015 and 2040. Of this waste stream, only 42% of the expected materials (by weight) is currently recycled in the U.S., including metals such as aluminum, cobalt, copper, nickel, and steel. Another 10% of the projected EV battery waste stream (by weight) includes two high value materials that are currently not recycled at a significant rate: lithium and manganese. The remaining fraction of this waste stream will include materials with low recycling potential, for which safe disposal routes must be identified. Results also indicate that because of the potential "lifespan mismatch" between battery packs and the vehicles in which they are used, batteries with high reuse potential may also be entering the waste stream. As such, a robust end-of-life battery management system must include an increase in reuse avenues, expanded recycling capacity, and ultimate disposal routes that minimize risk to human and environmental health.

1. Introduction

Lithium-ion batteries (LIBs) have emerged as a promising energy storage solution for electric vehicles (EVs) and renewable energy systems, but their potential environmental tradeoffs are not well characterized. Although recent work has focused on supply side issues, such as lithium availability, key uncertainties surround the emergence and management of these batteries in the waste stream and the ability of domestic recycling infrastructure to recover scarce and valuable materials from a highly variable mix of discarded batteries. A proactive approach is required to prevent unanticipated environmental impacts of end-of-life (EOL) battery generation associated with forecast growth in electric vehicle deployment.

Several agencies have predicted widespread diffusion of electric-drive vehicles in the future, both in the U.S. and at a global level. Forecasts of future EV sales (Fig. 1) have been produced by the U.S. Energy Information Administration (EIA, 2012), J.D. Power and Associates (Humphrey et al., 2010), Credit Suisse (Jobin et al., 2009), International Energy Agency [IEA] (2011), Deutsche Bank (Watabe and Mori, 2011), Deloitte Consulting (Giffi et al., 2010), Lazard Capital Markets (Shrestha et al., 2010) and Morgan Stanley (Steinmetz and Shankar, 2008). The range of deployment scenarios by these agencies vary significantly across parameters (economic growth, oil price, proposed Corporate Average Fuel Economy [CAFE] standards, battery technology etc.), and indicate anywhere between 0.45 million and 4 million EVs sold in the United States in 2020 (Fig. 1(a)) and international sales ranging between 5.2 million and 19.8 million in the same time frame (Fig. 1(b)). Powering these vehicles will clearly require a large scale deployment of energy storage systems (Gaines and Nelson, 2010; Gruber et al., 2011; Kushnir and Sandén, 2012).

This rapid growth in LIB demand comes with its own sustainability tradeoffs: as replacements for nickel metal hydrides (NiMH), LIBs reduce demand for rare earth metals but increase consumption of lithium, cobalt, manganese, and nickel (Alonso et al., 2012; Gruber et al., 2011). Several studies have investigated the implications of EV penetration on material demand, particularly lithium

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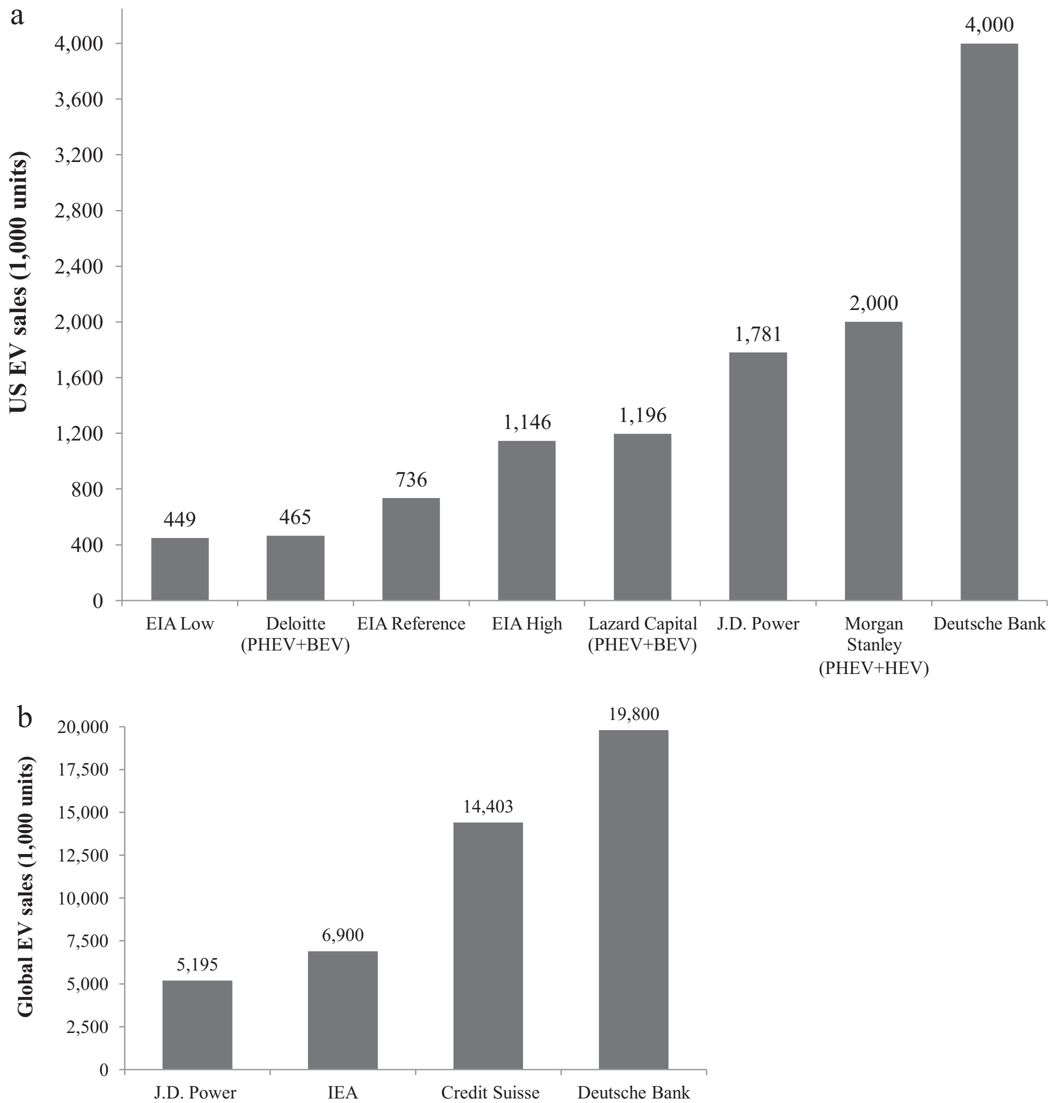

Fig. 1. (a) EV sales forecast-2020 (U.S.). (b) EV sales forecast-2020 (Global). Literature references for each sales forecast provided in the main text.

(Gaines and Nelson, 2010; Gruber et al., 2011; Grosjean et al., 2012; Kushnir and Sandén, 2012; Yaksic and Tilton, 2009). Though concerns over lithium scarcity in the long-term have been lessened by reassuring results from such studies, there may still be future challenges for the U.S. to access world lithium resources. A large portion of lithium deposits are found in only a few countries of the world, with the U.S. accounting for only 0.3% of current lithium reserves (USGS, 2012) and about 3.7% of the world lithium reserve base (USGS, 2009). Trade embargoes or political instability in the future may impact the U.S. EV and LIB industries, as many lithium-supplying countries are already politically volatile. Furthermore, cobalt, manganese and nickel, which are major inputs to the lithium-ion battery industry, are not significantly mined in the U.S., requiring primary dependence on imported supplies (USGS, 2012). Hence, EV battery recycling may contribute to a stable domestic supply chain for these critical materials. Currently, most EOL batteries from consumer electronics contain high levels of cobalt, a metal whose high economic value catalyzes current LIB recycling systems, but the trajectory of battery technology could result in introduction of different material and value streams, which may change the economic and policy implications of battery recycling (Wang et al., 2013,2014).

Clearly, a better understanding of the ultimate management and fate of batteries in the waste stream is required, but such an analysis is complicated by key uncertainties, including the expected timing and volumes of batteries reaching their end of life; the quality, concentrations, and variability of specific materials contained in spent batteries; and the capacity for recycling systems to recover scarce and valuable materials from a highly variable battery waste stream. While the lag in deploying EV technologies may suggest that battery waste will not be a priority for several years, "lessons learned" from our current sub-optimal management of electronic waste show the perils of introducing complex products without proactive development of a waste management system. In the case of electronic waste, low EOL value, difficulty recovering valuable materials and insufficient domestic infrastructure has lead to exploitation of developing countries and loss of valuable material resources (Babbitt et al., 2011; Williams et al., 2008; Widmer et al., 2005; Wang and Gaustad, 2012). Since many of these factors are similar to LIBs, avoiding negative environmental, economic, and social outcomes at EOL requires a more proactive approach in planning for this new waste stream.

As a step towards addressing EOL LIB management, this paper applies a scenario-driven material flow analysis (MFA) to project

the potential volume and timing of lithium-ion batteries entering the waste stream as a result of their forecasted deployment in electric vehicles. Towards this objective, the number of EV LIB units entering the waste stream as well as the mass of battery cells in that stream is estimated on an annual basis between years 2015 and 2040 for three different scenarios. To estimate recycling potential and waste management needs of EOL EV LIBs in the future, this paper also aims to characterize the materials that would be present in the EV battery waste stream on the basis of their recyclability and their commodity value under different technology trajectories of battery chemistry and form factor. Furthermore, this MFA model also seeks to characterize the potential for diverting EV batteries from this waste stream into reuse applications depending on the remaining battery life.

MFA is a well-established method for investigating the material, energy and environmental implications of commodity products (Oguchi et al., 2008; Yoshida et al., 2009; Chang et al., 2009; Steubing et al., 2010). While some MFA studies have addressed issues related to LIBs, the existing literature focuses on analyzing the stock and flows of laptops and cell phones batteries (Chang et al., 2009), tracking flows of cobalt (Harper et al., 2012), and assessing supply and demand for lithium due to EV technology (e.g., Gaines and Nelson, 2010; Grosjean et al., 2012; Kushnir and Sandén, 2012). To our knowledge, no study has yet applied MFA to fully model future outflows of batteries from EV systems.

Because such an analysis is complicated by significant uncertainty about technology adoption and performance, this MFA is also informed by approaches used in previous studies to develop scenario-based MFA for materials ranging from steel (Park et al., 2011; Pauliuk et al., 2011; Michaelis and Jackson, 2000) to electronic waste (Steubing et al., 2010; Kang and Schoenung, 2006; Streicher-Porte et al., 2005). From a methodological standpoint, this paper also highlights the uncertainties associated with conducting a scenario-based MFA of EV LIBs, as a means of establishing future research priorities that must be resolved as additional data and system parameters become available. Key variables addressed here include EV adoption dynamics, battery lifespan and constituent materials in lithium-ion cells.

2. Methods

A future oriented top-down material flow analysis (MFA) was conducted to estimate the volume of lithium-ion batteries projected to enter the waste stream in the near and long term future, after use in electric vehicles. MFA is a systematic assessment of the flows and stocks of materials within a defined temporal and spatial system (Brunner and Rechberger, 2004) that can be used to track the flow of a specific substance or of products within a system. In the top-down MFA methodology, the product inflows are determined from specific 'final goods' categories entering the system and the outflows are determined from discards, based on product lifespan, with the material stocks being inferred from these inflows and outflows (Graedel and Allenby, 2010). Here, the annual inflow of EV batteries was estimated from projected EV sales, and the annual outflow of batteries was calculated based on battery and vehicle lifespans. Given the significant uncertainty about future EV adoption rates and battery technologies, bounding scenarios were developed to forecast "low," "baseline," and "high" projections of future waste battery outflows and their attendant material implications. Key differences among these scenarios stemmed from variability in EV sales projections, battery lifespan distribution and parameters governing number of cells per battery pack, which will be discussed in the subsequent sections.

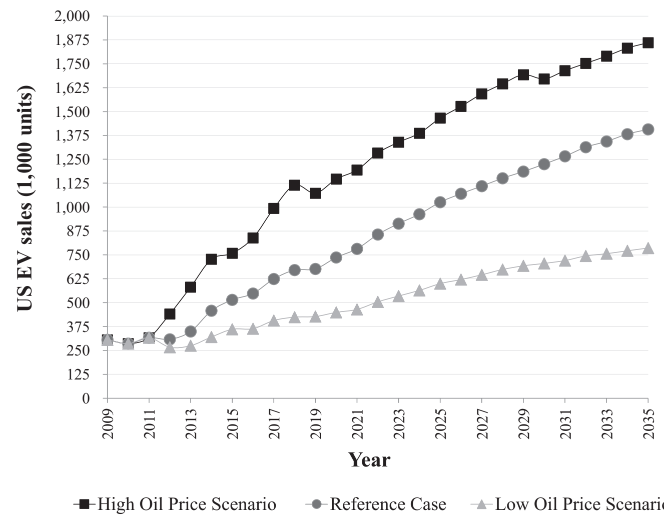

Fig. 2. Energy Information Administration (2012) EV sales forecasts.

2.1. Model formulation

The EV battery MFA model was implemented in three sequential steps, each described in more detail in the following sections:

(1) Material flow analysis to estimate the waste flows of entire EV battery packs.
(2) Estimation of individual lithium-ion cells in the EV battery pack waste stream.
(3) Assessment of specific materials comprising each cell within the EV battery waste stream.

Furthermore, based on the material and mass composition of the EV battery waste stream, the economic value of the waste stream was estimated on an annual basis.

2.1.1. Material flow analysis to estimate the flows of waste EV battery packs

The first part of the model calculated the number of lithium-ion EV battery packs entering the U.S. waste stream on an annual basis from years 2015 to 2040. This time period was chosen based on available data from the Department of Energy on both near- and long-term EV deployment projections. The annual inflow of EV batteries was estimated from EV sales forecasts, and the annual outflow of waste batteries was determined based on the battery lifespans once they entered vehicle use, as well as the lifespan of the EV itself.

2.1.1.1. EV sales forecast. U.S. level EV sales forecasts were obtained from the Light Duty Vehicle (LDV) Sales Projections through the year 2035 provided by the U.S. Energy Information Administration [EIA] (EIA, 2012). Three types of electric-drive vehicles were considered in the EIA LDV sales forecasts: hybrid- [HEVs], plug-in hybrid- [PHEVs] (10 miles and 40 miles ranges) and all-electric or battery electric vehicles [BEVs]. For the baseline scenario, the EIA Annual Energy Outlook "reference case" LDV sales projections were used. The reference case used in EIA projections is a baseline scenario assuming business-as-usual with current laws and regulations being the same across the timeline of the projections (EIA, 2012). The low and high scenarios reflect EIA forecasts that consider low and high oil prices, respectively. These forecasts are shown in Fig. 2.

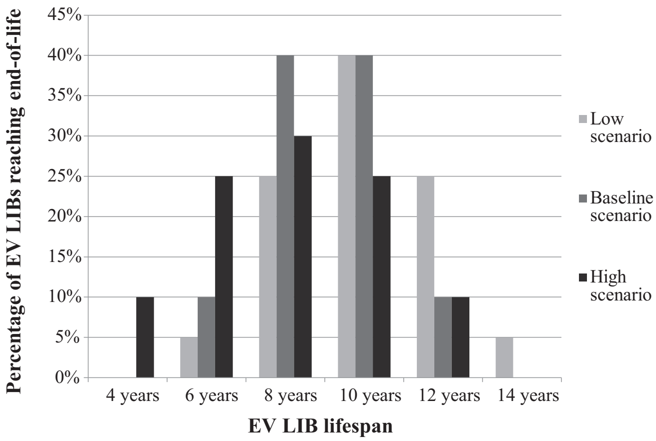

Fig. 3. Truncated lifespan distribution of EV batteries for three scenarios.

2.1.1.2. Battery lifespan. The lifespan or service life of a lithium-ion battery can be expressed either in terms of its cycle life or its calendar life. Cycle life is defined as the number of charge-discharge cycles the battery can undergo before failing to meet specific performance criteria. Calendar life on the other hand is defined as the length of time a battery can be stored with minimal discharges before capacity diminishes. In general, a battery is considered to have reached its end of life in EV application when it reaches about 80% of its original capacity (Williams and Lipman, 2010). EV battery lifespan is highly uncertain and dependent on many factors which are still poorly understood. Marano et al. (2009) indicates that lithium-ion batteries usually have a calendar life of 10 years, subject to favorable operating conditions that avoid overcharging, aggressive driving leading to rapid discharge and more frequent charging, and operation at high temperatures. Most previous studies have assumed a fixed EV battery lifespan of either 8 or10 years (Gruber et al., 2011; Yaksic and Tilton, 2009; Harper et al., 2012), which is consistent with the length of many vehicle manufacturers' warranty terms. However, some literature indicates lower lifespan of about 5 years for EV LIBs (Anderman, 2007). As per Dinger et al. (2010), EV battery lifespan could be anywhere between 5 and 10 years, while Nemry et al. (2009) assume a lifespan of 10–15 years. Significant research efforts are aimed toward achieving lifespans of up to 15 years for EV batteries (Kalhammer et al., 2009; Chalk and Miller, 2006).

Applying a lifespan distribution to determine the EV-LIB outflows would address the fact that the lifespan of a battery would depend on its usage and charging patterns, which vary from user to user. Assuming that electric vehicles are charged 1.5 times per week, the U.S. Department of Energy (DOE, 2010) predicted that the calendar life of a typical EV battery would increase from 4 years in 2009 to 14 years in 2015 owing to the ongoing innovation in this field. Hence, the different scenarios in the model have considered battery lifespans ranging from 4 to14 years. Rather than a single point estimate, a lifespan distribution (Fig. 3) was applied to model a more realistic scenario, taking into consideration early battery failures as well as batteries surviving for more than 10 years. Since a lifespan distribution of EV LIBs is not yet established, this technology being in its early stages of adoption, a truncated normal distribution of EV LIB lifespan has been used in the three scenarios (with a mean lifespan of 8 to 10 years). The variation in assumed battery lifespan distribution among the three scenarios not only indicate the uncertainty in the lifespan of EV LIBs but also highlight that the volume of EV battery waste stream would be dependent on battery lifespan to a certain extent. For instance, in the low scenario, 70% of EV LIBs have been assumed to have lifespan exceeding 8 years, whereas this percentage is 50% and 35% respectively for batteries in the baseline and high scenarios respectively. In spite of these variations, the distributions selected result in a majority of EV LIBs used in EVs modeled as having a lifespan in the range of 8–10 years, consistent with warranty terms and recent literature.

Similar to EV batteries, the lifespan of electric vehicles too would follow a distribution which may be even wider than that for batteries, depending on early vehicle failure or car crashes as well as extended life through multiple resales. However, to keep this initial MFA model tractable, the EV lifespan has been fixed. In general, traditional vehicle lifespan assumptions vary across studies in the range of 10–16 years (Huang et al., 2011; Greene and DeCicco, 2000; Lemp and Kockelman, 2008; Greene et al., 2005; Kumar and Sutherland, 2008). Only limited information is available on electric vehicle lifespan, but this parameter is modeled as 10 years in a recent study by Gruber et al. (2011). While we recognize the uncertainty associated with lifespan and the need for future work in this area, this MFA model assumed a moderate, fixed EV lifespan of 10 years as a starting point for analysis, with sensitivity analysis on a 16 year EV lifespan shown in section S7.3 of the supplementary information (SI).

The lifespan distribution shown in Fig. 3, contrasted against the vehicle life, raises an important point: there will likely be a "mismatch" between vehicle and battery lifespans. Some batteries entering use in a given year would likely reach the end of their life before the vehicles in which they are used. These vehicles then need new batteries to continue operation in subsequent years. On the other hand, if a vehicle were to reach the end of its life before its batteries, it is assumed that the battery would not be refitted into a new vehicle (although it may be reused in other applications) (Williams and Lipman, 2010; Cready et al., 2003). Thus, batteries entering the waste stream at any given time can be loosely classified into two types:

Type 1 EOL EV batteries are those that have reached their end-of-lives in EV application due to capacity fade, either before or coinciding with the vehicles' end of life. In general, an EV battery has 70–80% of its original capacity intact once it reaches the end of its utility for EV applications (Neubauer and Pesaran, 2011). Though insufficient for automotive use, there is some potential that these batteries can be reused in off-grid and grid-based stationary energy storage applications instead of entering the waste stream (Neubauer and Pesaran, 2011; Williams and Lipman, 2010; Cready et al., 2003).*Type 2 EOL EV batteries* are those found in vehicles that reach their end-of-lives before their batteries, which is likely the case in early vehicle failure or crash or if a vehicle has a battery replacement later in its useful life. This set of non-EOL EV batteries could technically still meet the criteria for reuse in EVs, but actual reuse in this manner is unlikely, given concern about reliability and technical compatibility of "pre-aged" batteries (Cready et al., 2003; Burke, 2009). These batteries may have high potential for other reuse markets, like those described above. The distinction between these two battery types is intended to indicate the potential for diverting batteries from the waste stream into reuse applications.

2.1.1.3. Lithium-ion battery use in hybrid electric vehicles. Currently most HEVs on the market use nickel metal hydride [NiMH] batteries, rather than lithium-ion, and NiMH batteries would continue to be a feasible option for HEV for several years (Frost and Sullivan, 2009). However, it is predicted that lithium-ion batteries' share of the HEV market would grow and eventually surpass NiMH usage between 2018 and 2025 (Jobin et al., 2009; Madani, 2009; Fu, 2009). Estimates from a Credit Suisse report prepared by Jobin et al. (2009) were applied to the scenarios used here, as their study provided both conservative and optimistic estimates for HEV lithium-ion battery adoption. The high and baseline scenarios started from the Credit Suisse bottom-up estimates, which were optimistic towards rapid LIB adoption in HEV (Jobin et al., 2009), leading to an

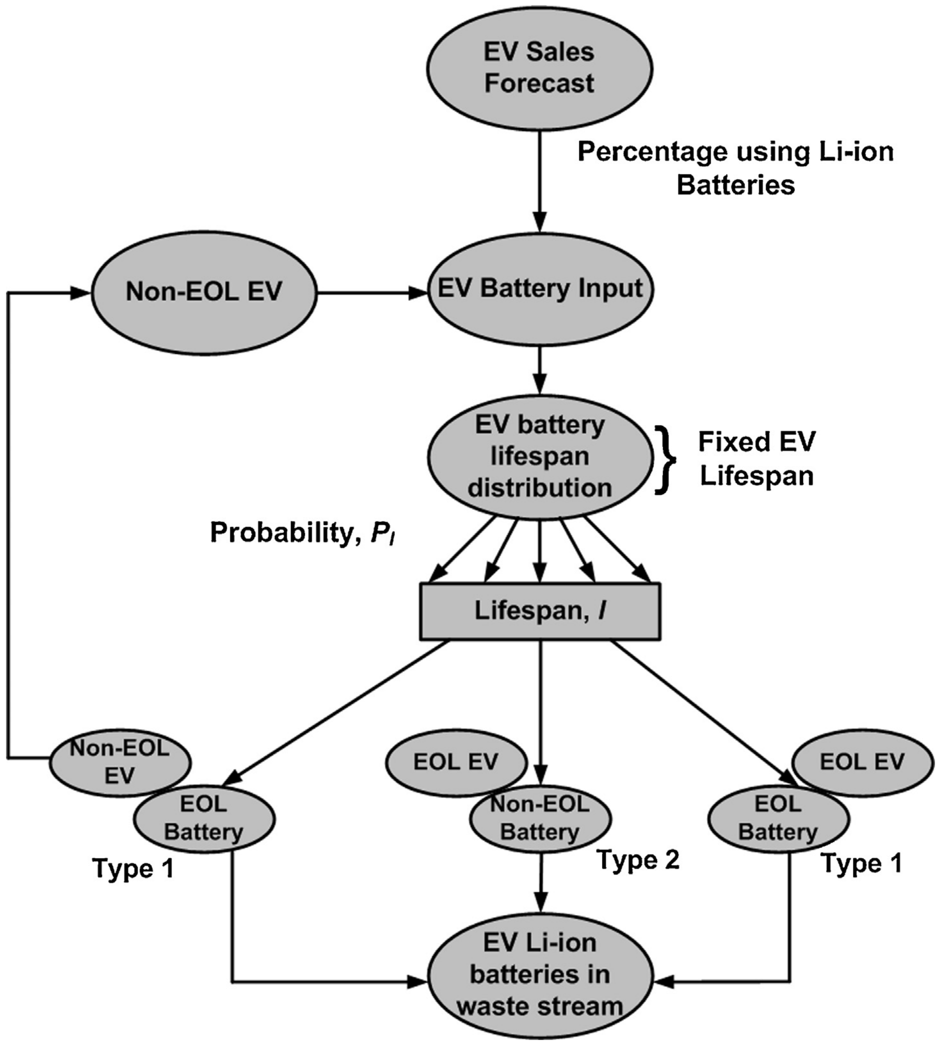

Fig. 4. Conceptual basis of estimating future EV Li-ion battery outflows.

assumption that all HEVs use lithium-ion batteries by year 2015 and 2025, respectively in these two scenarios. The Credit Suisse top-down estimate for HEV lithium-ion battery adoption was used for the low scenario, with an assumption that 100 percent of HEVs would not rely on lithium-ion batteries until the year 2032. Details about HEV lithium-ion battery adoption is provided in the SI.

2.1.1.4. Estimation of EV battery pack outflows. Considering the sales and lifespan assumptions stated above, Fig. 4 illustrates the conceptual basis of estimating EV battery outflows by this model.

The number of new LIBs entering EV use in any year t would depend on EV sales in year t, as well as the number of non-EOL EVs which would require a replacement battery in that year (Fig. 4). Here, it was assumed that all non-EOL EVs would use a replacement battery, while we do recognize that realistically, all vehicles may not be put back into use due to high replacement battery cost or damages due to automotive accidents. The lifespan distribution was based on "P_l", the percentage of batteries sold in any given year to have a useful life of l years in EV application, which varied based on the scenario (Fig. 3). K_t, the total number of lithium-ion battery packs entering use in EVs in year t was determined as follows:

$$K_t = \sum_i \left(S_{i,t} + W_{i,t}\right) \quad (1)$$

$S_{i,t}$ = sales of new EVs of type i that use LIBs in year t and $W_{i,t}$ = non-EOL EVs of type i requiring a replacement LIB in year t.

The above relationship was distinguished by "i" vehicle types: BEV, PHEV10, PHEV40, and HEV (the percentage of HEVs that use lithium-ion batteries). The number of batteries entering new EVs ($S_{i,t}$) was determined by the sales forecast for that year, as described in a previous section. The non-EOL EVs requiring a replacement battery ($W_{i,t}$) was based on the scenario-specific cases of first-use batteries with a shorter lifespan than the vehicles in which they were used. In cases of extreme "lifespan mismatch," vehicles with very long lifespans paired with batteries with very short lifespans may require two battery replacements. Hence, for a given year, t,

$$W_{i,t} = \sum_i \sum_l P_l \times (S_{i,(t-l)} + W_{i,(t-l)}) \quad (2)$$

l = EV battery lifespan, s.t. l < EV lifespan, P_l = percentage of EV LIBs sold in any given year to have a useful life of l years in EV application, $S_{i,(t-l)}$ = sales of new EVs of type i in year $(t-l)$, and $W_{i,(t-l)}$ = non-EOL EVs of type i requiring a replacement battery in year $(t-l)$.

Thus, the number of EV LIB packs entering the waste stream (B) in a given year t after an l year lifespan is expressed as:

$$B_t = \sum_i \sum_l P_l \times (K_{i,(t-l)}) \quad (3)$$

Table 1
Vehicle consumption rate, battery efficiency, percent available energy and EV battery energy storage for the three scenarios.

Scenario	Vehicle consumption rate (Wh/mile)	Battery efficiency	Battery available energy (%)			EV battery energy storage (kWh)			
			BEV	PHEV	HEV	BEV	PHEV10	PHEV40	HEV
Low	250	95%	90%	80%	30%	29	3.3	13	3.5
Baseline	300	90%	85%	75%	25%	39	4.4	18	5.3
High	350	85%	80%	70%	20%	51	5.9	24	8.2

$K_{i,(t-l)}$ = total number of LIB packs entering use in EVs of type i in year $(t-l)$.

2.1.2. Estimation of individual lithium-ion cells in the EV battery pack waste stream

The approach described thus far focused on total battery packs, which each may contain a varied number and type of cells, depending on technical specifications such as EV type and cathode chemistry. Next, we estimated the number of lithium-ion cells in this EV battery waste stream, for a given year t as

$$N_t = B_t \times \sum_i \sum_j (PE_{i,t} \times PC_j \times D_{i,j}) \quad (4)$$

B_t = number of LIB packs in EV battery waste stream in year t, i = EV type (BEV, PHEV10, PHEV40, HEV), j = LIB cathode chemistry, $PE_{i,t}$ = percentage of waste LIB packs belonging to EV type i in year t, PC_j = percentage of LIBs of battery chemistry j in EV battery waste stream, and $D_{i,j}$ = number of cells per LIB pack, specific to EV type and cathode chemistry.

Parameters B_t and $PE_{i,t}$ change with time as well as with the scenario under consideration as they are functions of annual EV sales within a given scenario. On the other hand, PC_j and $D_{i,j}$ were assumed constant with time, though $D_{i,j}$ does vary across the scenarios as shown in Table S3.2.

The number of battery packs in a given year t (B_t) was obtained from the EV battery MFA results discussed in the previous section. $PE_{i,t}$, percentage of waste batteries belonging to a given EV type in year t was based on the relative prevalence of each type of EV sold, and thus entering the waste stream. Four prevalent lithium-ion cathode chemistries (i.e. j) were considered, namely, lithium cobalt oxide ($LiCoO_2$), lithium manganese oxide ($LiMn_2O_4$), lithium iron phosphate ($LiFePO_4$) and lithium nickel cobalt manganese (NCM) oxide, all having 18650 form factor cells (cylindrical cells with 18 mm diameter and 65 mm length). The selection of this form factor was based on data availability, with the recognition that results may change with alternative form factors, like the prismatic cells, expected to be used in most EVs. Sensitivity analysis was conducted on this assumption as described in following sections. While the current LIB waste stream is almost entirely made up of consumer electronic batteries, which typically contain 100% $LiCoO_2$ cathode chemistry (Wang et al., 2014), the distribution of cathode chemistries assumed in this paper (i.e. PC_j) for all three scenarios is 10% $LiCoO_2$, 30% $LiMn_2O_4$, 30% $LiFePO_4$, and 30% NCM. This distribution was selected to reflect that all three latter cathode chemistries are likely candidates to replace the existing lithium cobalt oxide based batteries for EV application, aside from limited application as in the case of Tesla vehicles. The number of cells per battery pack for a given EV type using a given battery chemistry ($D_{i,j}$) varied with the scenario under consideration as described in the following section.

2.1.2.1. Determination of number of cells per battery pack ($D_{i,j}$). The number of cells per LIB pack for a given EV type and a given battery chemistry ($D_{i,j}$) was estimated from the energy storage capacity of the EV battery pack ($Epack_i$), dependent on the EV type i and the energy storage of individual cells ($Ecell_j$), dependent on the cell cathode chemistry j and was estimated as follows:

$$D_{i,j} = \left(\frac{Epack_i(\mathrm{Wh})}{Ecell_j(\mathrm{Wh})} \right) \quad (5)$$

2.1.2.2. Battery energy storage ($Epack_i$). The battery pack energy storage ($Epack_i$) depends on the EV type and its associated electric range, as well as other parameters like vehicle consumption rate and percent battery efficiency and available energy:

$$Epack_i = \frac{(R_i \times C)}{(\eta \times A_i)} \quad (6)$$

R_i = electric range of EV type i (miles), C = electric vehicle consumption rate (Wh/miles), η = percent efficiency of EV LIB, and A_i = percent available energy of the total EV LIB energy for a given EV type i

While the vehicle electric range remains constant with each scenario, the other three parameters in Eq. (6) will vary over the three scenarios:

(1) *Vehicle electric range (R_i)*: The electric ranges for the three EV types were determined based on EIA (2012) and Gaines and Nelson (2010) and were kept fixed across all scenarios. The BEVs were assumed to have 100 miles electric range, the HEVs were assumed to have 4 miles electric range. In case of PHEVs, both 10 and 40 miles electric ranges were considered.
(2) *Vehicle consumption rate (C)*: The consumption rate of an electric vehicle can be defined as the electrical energy consumed per mile of travel. Table S5.1 in the supplementary information lists the energy consumption rates of electric vehicle models in the recent years (according to EPA tests), and assumptions documented in the literature. Based on these values, the consumption rate of EVs was assumed to be 250 Wh/mile, 300 Wh/mile and 350 Wh/mile for the low, baseline and high scenarios, respectively.
(3) *Battery efficiency (η)*: The overall energy stored by the battery available for electric vehicle application depends on the energy efficiency of the battery, so there is an inverse relationship between efficiency and number of cells. The battery efficiency determines the amount of energy taken out during discharge after it was initially charged. The most common energy efficiency value for lithium-ion batteries reported in literature is 90% (Van den Bossche et al., 2006; Gondelach, 2010; Shiau et al., 2009; Karden et al., 2007; Tanaka et al., 2001; Matheys et al., 2008). For the Tesla Roadster BEV, the efficiency of the charge-discharge cycle of lithium-ion batteries was reported to be approximately 86% (Eberhard and Tarpenning, 2006). Campanari et al. (2009) have assumed a 92% efficiency of lithium-ion batteries used in electric vehicles. According to Rydh and Sandén (2005), the efficiency of lithium-ion batteries can lie anywhere between 85% and 95%. Hence, the EV battery efficiency was assumed to be 95%, 90%, and 85%, respectively for the low, baseline, and high scenarios (see Table 1).
(4) *Available energy of EV battery (A_i)*: The available energy of an EV battery is typically less than the total energy stored because

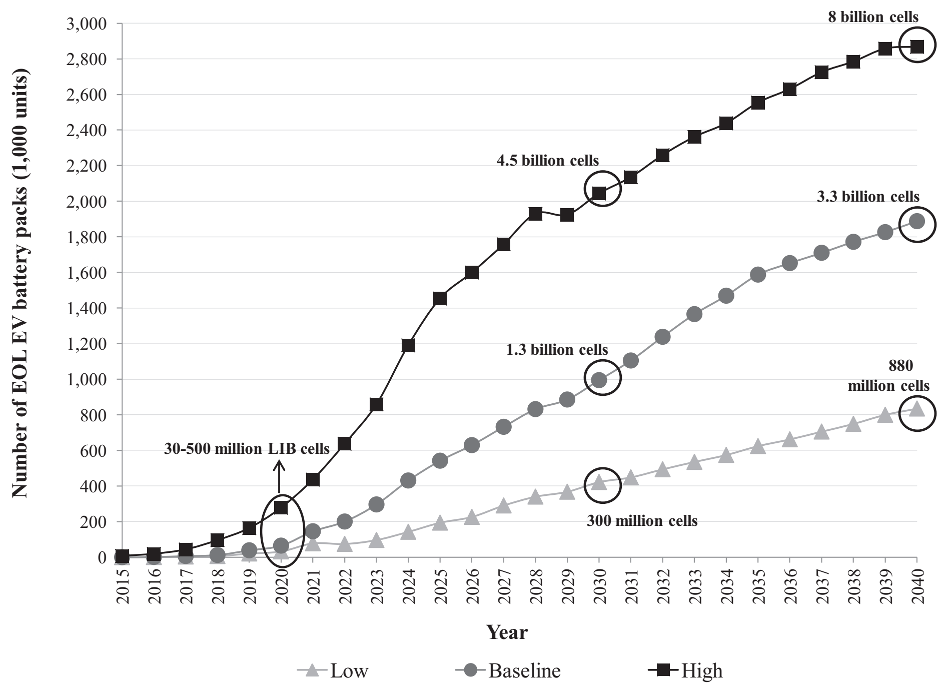

Fig. 5. End-of-life Li-ion batteries generated from EV applications annually between 2015 and 2040.

the depth of discharge is restricted to preserve battery life and for safety purposes (Axsen et al., 2008). According to Srinivasan (2008), the available energy of a HEV battery is 20-30% of its total energy, while for a BEV or PHEV battery it could be as high as 70–80%. As per the Argonne National Laboratory [ANL] (2012) BatPaC model, the energy utilized by a HEV battery is 25% of the total energy, while it is 70–75% and 85–90% for a PHEV and BEV battery respectively. Based on these ranges, assumptions for the available energy percentage of the total battery energy for each of the vehicle type and for the three scenarios are as shown in Table 1, which also includes assumptions for the vehicle consumption rate, battery efficiency and available energy for the three scenarios, and the calculation of EV battery energy storage based on these factors.

2.1.2.3. Cell energy storage ($Ecell_j$). The energy storage of 18650 cells for the four battery chemistries considered was obtained as the product of cell capacity and the nominal or average cell voltage as described in section S3 in the SI. The cell capacity (mAh) was estimated as the product of the cathode mass and the specific capacity (mAh/g) of lithium-ion cells for each of the four cell chemistries considered in the model. The specific capacity of the lithium-ion cells was obtained from Dahn and Erlich (2011). The cathode mass of each of the cell types was estimated from their respective bill of materials. The cell energy storage of each of the lithium-ion cell types was assumed to be constant with time as well as across the three scenarios. Using the approach described above, the final input to the MFA model pertaining to number of cells per LIB pack was determined (summarized in Table S3.2 in SI, which distinguishes across scenarios, vehicle types, and cathode chemistries).

2.1.3. Assessment of specific materials comprising each cell within the EV battery waste stream

In the final stage of modeling, the specific materials contained in the battery cells were taken into account. Based on B_t, the total number of waste LIB packs in year t, the percentage of waste LIBs belonging to EV type i in year t ($PE_{i,t}$), the percentage of battery chemistry j in EV battery waste stream (PC_j), and the number of cells per battery pack for EV type i and battery chemistry type j ($D_{i,j}$), the amount of any material y present in the EV battery waste stream for a given year t was estimated as,

$$MO_{y,t} = B_t \times \sum_i \sum_j (PE_{i,t} \times PC_j \times D_{i,j} \times m_{y,j}) \quad (7)$$

$m_{y,j}$ = mass of a given material y (aluminum, copper, lithium etc.) in a lithium-ion cell of cathode chemistry j.

The variable $m_{y,j}$ was obtained from the bill of materials of lithium-ion cells of the four cathode chemistries from the disassembly of 18650 lithium-ion cells (Wang et al., 2013) and remained constant across the scenarios as well as with time (Table S4.1, SI). The other variables in this part of the model have been discussed in previous sections.

2.2. Economic value of materials in EV battery waste stream

The annual value of materials present in the EV battery waste stream was estimated using global spot prices (London Metal Exchange, 2012; Shanghai Metals Market, 2012) and USGS (2012) commodity values of LIB materials (Table S9.1, SI). This estimation only included currently recycled materials (aluminum, cobalt, copper, nickel, steel and iron) as well as high value materials not currently recycled in the U.S. but with high potential for recovery in the future (lithium and manganese) to calculate the "maximum theoretical commodity value" of the EV battery waste stream. The future-oriented characterization of lithium and manganese as high value materials is based on several factors, including current LIB recycling efforts aimed at developing recovery processes for these materials (Paulino et al., 2008; Dunn et al., 2012; Zou et al., 2013; Yang et al., 2013), limited lithium and manganese resources in the U.S. and the resultant dependence on import of these metals (USGS, 2012), and the potential price rise of these metals with growing demand for EV LIBs. Manganese comprises about 20–25% of a typical lithium-ion cell (Wang et al., 2013; ANL, 2012) making these cells a viable source for recovery of manganese. Though lithium constitutes only 1–2% of the total cell mass of typical LIBs (Wang et al., 2013; ANL, 2012), considering an EV battery pack comprising

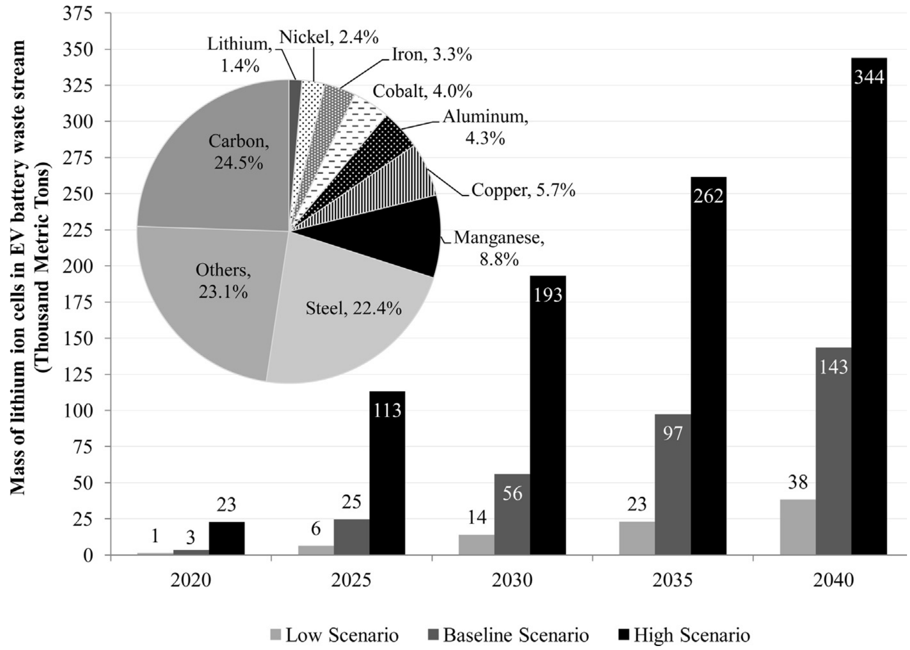

Fig. 6. Mass of Li-ion cells in EV battery waste stream. (In the pie-chart, "carbon" includes carbon black and graphite. "Others" include plastics, binders, electrolytes and other non-metals like phosphorus etc.)

of thousands of cells, the amount of lithium available for recovery would not be negligible. Although lithium carbonate is currently a lower cost input to LIB production (USGS, 2012), the forecast increase in lithium demand by 2020 (Jobin et al., 2009) and potential lag in supply (Kushnir and Sandén, 2012) may trigger lithium price rise in future. In fact, lithium spot prices of about $62/kg have been listed in the Shanghai Metals Market (2012).

Recycling efficiencies of materials and the collection rate of spent EV LIBs were not considered in estimating the commodity value of EV battery waste stream. Other materials in this waste stream that are unlikely to be recycled (graphite, electrolyte, plastics, etc.) were excluded from this valuation. The baseline scenario MFA results were used as basis for these economic estimations.

3. Results and discussion

3.1. Estimation of EOL battery packs from electric vehicles

Based on the parameters defined for each of the three scenarios, the number of EV LIB packs potentially entering the waste stream on an annual basis was estimated (Fig. 5).

While the three scenarios projected similar increase in EV battery waste flows in the U.S. during the first five years of the analysis, results quickly diverge due to differences in input sales and battery lifespans. As per the baseline, approximately 1.9 million LIB packs (each consisting of many cells) could be entering the waste stream annually by year 2040. However, considering the range from the most conservative to most extreme estimates, the waste stream could hypothetically fall anywhere between 0.83 and 2.87 million LIB packs per year by 2040. The cumulative baseline outflow of LIB packs between 2015 and 2040 (21 million packs) was approximately two and a half times greater than the total number of EV battery packs calculated in the "low" scenario (8.7 million packs) and about two times fewer than that of the "high" scenario (40 million packs). Of these LIB packs, between 27 and 35% would be coming from all-electric and plug-in hybrid electric vehicles, and the remaining 65–73% were estimated to be from hybrid electric vehicles, reflecting the projected sales of each vehicle type.

The battery waste flows were differentiated based on the "Type 1" and "Type 2" classifications of remaining life as described earlier (Table 2, also summarized in Fig. 7 for the baseline). Characterization of EOL batteries into these categories provides some indication of the volume of batteries with the highest potential for suitable reuse applications. For instance, type 2 EOL batteries still hypothetically have remaining EOL life, making them better suited for applications requiring high capacity. In each of the scenarios, Type 2 batteries represent a sizeable fraction, and despite current hesitance surrounding reuse in vehicles, the number of batteries expected suggests that "re-matching" Type 2 batteries with older vehicles or some other form of cascading use should be studied further.

3.2. Lithium-ion cells and attendant material flows in the EV battery waste stream

The EOL EV batteries generated on an annual basis would contain hundreds or even thousands of cells, each consisting of different metals, carbonaceous materials (carbon black and graphite) and other miscellaneous materials such as organic carbonates, lithium salts, binder, plastics, etc. Considering the parameters specified for the baseline scenario, approximately 3.3 billion individual lithium-ion cells may be entering the waste stream annually by 2040. By that point, the cumulative outflows between years 2015 and 2040 would be on the order of 30 billion cells requiring EOL management. The annual waste flows could be as low as 0.88 billion cells (low scenario) or as high as 8 billion cells (high scenario) per year by 2040.

Table 2
Percentage of Type 1 and Type 2 EOL EV batteries accrued in the waste stream between 2015 and 2040.

Scenario	Percentage of Type 1 EOL batteries	Percentage of Type 2 EOL batteries
Low	57%	43%
Baseline	63%	37%
High	62%	38%

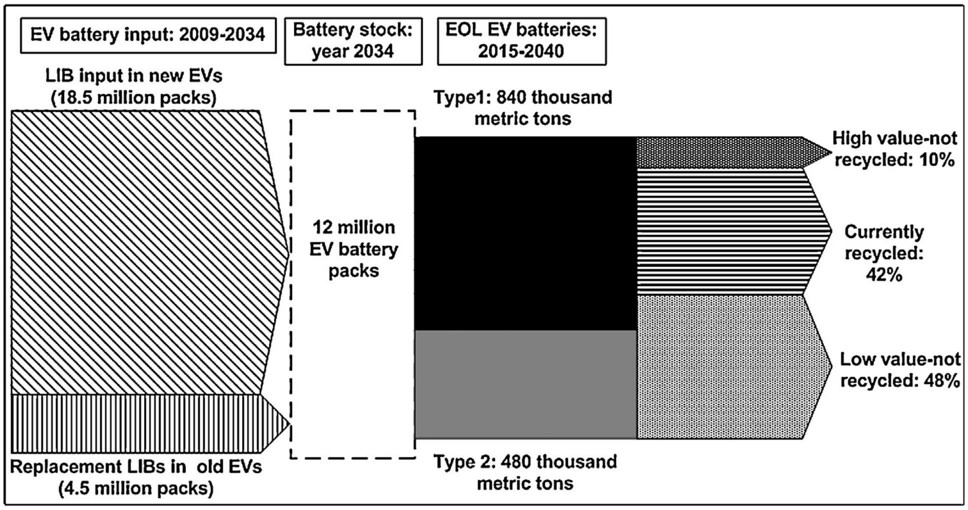

Fig. 7. EV battery material inflow and outflows-Baseline Scenario. Thickness of each bar corresponds to the relative mass of material in each category.

For the baseline scenario in year 2020, the LIB waste stream could contain approximately 3,400 metric tons of lithium-ion cells just from EV application, which is about 4 times the estimated collection volume of waste LIBs from consumer electronics in 2012 (Wang et al., 2014). In terms of the resultant material flows, the range of scenarios indicate a total EV battery waste stream between 0.33 and 4 million metric tons, with a baseline estimate of 1.3 million metric tons generated cumulatively between 2015 and 2040. Fig. 6 summarizes annual outflows of battery materials on five-year increments over the long and short term future (Extensive annual estimates of EOL EV LIB material outflows are provided in SI, section S6.3). In this initial estimate, the material-specific composition of that waste stream does not vary, and is also summarized in Fig. 6.

In comparison with the small body of recent literature on lithium demand for EVs, this MFA predicted relatively conservative outflows, even for the “high” scenario. For instance, Gaines and Nelson (2010) estimated a maximum waste flow of 20,000 metric tons of lithium in 2040 from “optimistic” EV deployment, a prediction about 10 times greater than our baseline scenario estimates for that year. To put our estimates in a global context, this study forecast cumulative lithium outflows between 2020 and 2040 between 4.5 thousand to 55 thousand metric tons for the U.S. On a global basis, Gruber et al. (2011) estimated 860 thousand metric tons of potentially recoverable lithium from EV batteries in the same time frame (with 100% recycling participation and 100% lithium recovery).

The disparity observed between the low and high scenario in this paper is indicative of the variability in estimates of EV sales, the battery lifespan (and resultant need for replacement battery packs, particularly in the high scenario) as well as the parameters determining the number of cells per EV battery pack. Even with these uncertainties, we can begin to analyze results further, using the baseline scenario as a focal point (to minimize the amount of data presented in the main text). The baseline scenario was further characterized on the basis of battery inputs, outputs, and material characteristics. Fig. 7 summarizes these characteristics for the cumulative input of LIBs in electric vehicles between 2009 and 2034, and the net EOL battery outflows between 2015 and 2040.

The majority (80%) of new batteries entering use would be paired with new EVs sold in the market, while the remaining 20% would be replacement batteries for existing in-use EVs (Fig. 7). About 63% of the batteries leaving use were “Type 1,” with no remaining life for EV applications; while the remaining 37% “Type 2” batteries may have been discarded before their true EV end-of-life. Previously shown materials analysis (Fig. 6) distinguishes different materials contained in the battery waste stream, but the ultimate fate of these materials depends on whether an infrastructure and market exist for their recycling back into productive use. The potential of each material to be recycled was determined by assessing current recycling practices and secondary markets available for these materials (USGS, 2012). Based on the potential to be recycled, the materials expected in the EV battery waste stream were categorized as currently recycled, high value-not recycled, and low value-not recycled materials.

Of the estimated battery outflows, low value materials, which are currently not being recycled and are not expected to be in the future, could constitute 48% of the EV battery waste stream and include graphite, carbon black, lithium hexafluorophosphate ($LiPF_6$), organic carbonates (such as ethylene carbonate or dimethyl carbonate), binder (polyvinylidene fluoride) and mixed plastics (polypropylene, polyethylene). Apart from plastics, none of these materials have a secondary market at present and it would not be economically viable to recover them from the waste stream. Moreover, as a mixed grade of plastics would be present in the battery waste stream, their recovery would not be likely due to high contamination. Because these materials are not suitable for recycling, infrastructure must be equipped to accommodate their introduction to landfills or other disposal routes. As such, relevant environmental and health impacts should be anticipated. The carbonaceous material present in the EV battery waste stream could raise concern in the future owing to their large quantity in the waste stream (Fig. 6) and knowledge of potential health impacts of particulate carbon (e.g., exposure to graphite dust can adversely affect respiratory system and pulmonary function (NIOSH, 2007)). The electrolyte used in LIBs can have toxicity concerns as well. For instance, the electrolyte salt $LiPF_6$ is a hygroscopic substance and in presence of moist air or water forms hydrogen fluoride gas (Archuleta, 1995), which has severe environmental risks and toxicity concerns (EPA, n.d.). Similarly, organic carbonates used as electrolyte solvents are mildly toxic, volatile and flammable compounds, producing toxic fumes on decomposition (Vimmerstedt et al., 1995). Environmental impacts of EV battery waste could also be a concern due to the non-biodegradability of binder and other plastics in lithium-ion cells.

Another 42% of the materials in the cumulative EV battery waste stream would include metals that are currently and expected to continue being recycled according to statistics from the USGS

(2012). This fraction includes aluminum (57,000 metric tons), cobalt (52,000 metric tons), copper (75,000 metric tons), nickel (32,000 metric tons), steel (295,000 metric tons) and iron (43,300 metric tons). These material masses are on a cumulative basis between 2015 and 2040. The remaining 10% of the EV battery waste stream would include two high value materials that are currently not recycled in the U.S., i.e., lithium (18,000 metric tons) and manganese (116,000 metric tons). Many of these metals (lithium, aluminum, nickel and cobalt) have high embodied energy when extracted from virgin resources (ecoinvent Centre, 2007). Hence, recycling of LIBs offers a dual benefit: avoided energy inputs for production of primary metals and potential economic revenue from material recovery, which is particularly high for cathode chemistries like Li_2CO_3 and NCM that contain 10–17% by weight of high-value cobalt.

3.3. Economic value of materials in EV battery waste stream

Considering the baseline scenario with a mix of lithium-ion cell chemistries, the total EV LIB waste stream may contain materials valued at approximately 3.8 billion USD on a cumulative basis between 2015 and 2040. This estimate is the maximum theoretical commodity value of the EV battery waste stream considering the potential value for materials that currently have recycling infrastructure in the U.S, and does not take into account material losses that would occur due to recycling inefficiencies. The total possible waste stream value would be increased by over 1.5 billion USD if Li and Mn are also included.

The actual economic value of the EV battery waste stream would depend on the LIB collection rates, the recovery rates of the various materials present in the stream, and the cost of recycling itself. Considering recent recycling efficiencies (see SI, Table S10.1), commodity value of approximately 3 billion USD could be obtained between 2015 and 2040 by recovery of metals such as aluminum, copper, nickel, cobalt, iron and steel assuming that 100% of batteries in the waste stream can be collected for recycling. Wang et al. (2014) analyzed the profitability of LIB recycling facilities for several possible future co-mingled LIB waste streams based on the current recycling efficiency of materials in LIBs: the potential value from recycling one metric ton of LIBs ranged from \$860 for $LiMn_2O_4$ cathode batteries to \$8,900 for $LiCoO_2$ cathode batteries. Continued development of advanced separation processes could increase the recycling efficiencies of materials present in EV LIBs and hence the economic motivation for recovering materials from these batteries. For example, a 10% improvement over current recycling efficiency for cobalt could raise the recycling revenue by 9% for cobalt based LIBs while a 10% improvement in copper recycling efficiency would only improve revenue from LIB recycling by 1–5%, depending on the cathode chemistry (Wang et al., 2014).

The materials potentially recoverable by EV LIB recycling could be used as inputs to the parent battery industry, as this sector is predicted to become more resource intensive as vehicle deployment increases. Increasing availability of secondary material sources would reduce U.S. dependency on foreign resources in the long run. Gaines and Nelson (2010) estimated that recycling LIBs could meet almost 50% of the lithium required for battery production in the U.S. by 2040. However, the recyclability of the EV battery waste stream and hence, the economic gains from battery recycling is likely to depend on the battery technology prevalent in the future in terms of cathode chemistry as well as the form factor of the lithium-ion cells used in these batteries. At present there is significant uncertainty in this domain, which is analyzed in the subsequent sections, along with the uncertainty due to differences in MFA parameters such as EV sales, battery lifespan and number of cells per EV battery pack.

3.4. Uncertainty analysis

3.4.1. EV sales and battery lifespan

It was established by the range of scenario results that the volume of LIBs in the EV battery waste stream would be highly dependent on EV sales and the actual battery lifespan. The sales of electric vehicles will depend on a number of factors in the mid- and long-term, such as oil prices, battery and vehicle cost, EV and battery technology, government subsidies, policies and regulations etc. (EIA, 2012). The lifespan of lithium-ion batteries in EV application will depend on battery technology progress as well as usage patterns at the consumer level. Long battery lifespans would have a two-fold benefit, first reducing the need for a second (or even third) replacement battery and thereby reducing the cost of ownership of electric vehicles, and second, raising the potential for post-EV battery reuse, which can also defray costs across the battery life cycle.

Uncertainty analysis was performed to tease apart the role of sales and lifespan parameters on LIB waste flows, by holding one parameter constant and varying the other (SI section S7.1). When the EV sales estimates are held constant at the baseline level, cumulative (2015–2040) LIB outflows increased 16% or decreased 15% from the shortest lifespan ("high" scenario) to the longest ("low" scenario). On the other hand, when the baseline EV LIB lifespan distribution was combined with the high and low scenario EV sales figures, cumulative (2015–2040) outflows of EOL packs could increase by as much as 62% and decrease by 52%, respectively. It is evident that even though the battery lifespan distribution plays a role in influencing the volume of EV battery waste stream, electric vehicle sales will be the governing factor influencing EV LIB waste flows in the future.

When a longer EV lifespan of 16 years was tested for model sensitivity, the cumulative (2015–2040) outflows of these batteries into the waste stream changed by less than 2%, although the annual waste stream volumes varied, as shown in SI section S7.3. Further, the percentage of Type 1 and Type 2 EV batteries also changed with increasing lifespan of EVs. For instance, when a longer EV lifespan of 16 years was assumed for the baseline scenario, the percentage of Type 2 EV batteries estimated to accrue in the waste stream between 2015 and 2040 reduced from 37% to 23% (detailed analysis in SI section S7.3).

3.4.2. Cell and battery energy storage and battery pack components

The energy storage of the battery pack also plays an important role in determining the amount of cells per pack, and thus the materials present in the EV battery waste stream. A sensitivity analysis was conducted wherein the baseline cumulative outflow of EOL EV battery packs was held constant while the estimated EV battery pack energy assumptions were varied between the low and high scenarios (SI, section S7.2). The resulting estimates of total material mass of the EV battery waste stream decreased by up to 29% or increased by up to 40% when the low or high scenario cells per pack assumptions were applied, as compared to baseline flows.

Hence, it follows that battery and EV technology (in terms of electric miles and vehicle mileage) would play a major role in governing flows of EV batteries in the waste stream. Throughout the EV sales forecast timeline considered in the model (2009–2034), the same battery chemistries were assumed and the energy storage by individual cells was held constant. According to Srinivasan (2008), the energy density of lithium-ion batteries has been increasing at the rate of approximately 5% per year over the last one and a half decade. The average energy density of a typical 18650 cell is approximately 200 Wh/kg (Howard and Spotnitz, 2007). According to Srinivasan and Lipp (2003), when lithium-ion batteries were introduced in the early 1990s, this number was around 90 Wh/kg.

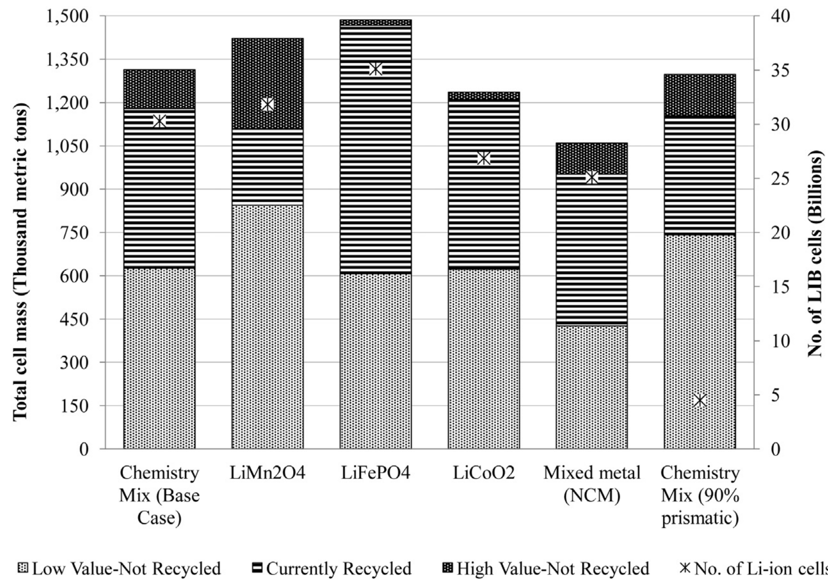

Fig. 8. Recyclability of EV battery waste stream under different scenarios of Li-ion cell cathode chemistry and form factor (cumulative flows from 2015 to 2040).

In other words, technological innovation has doubled the energy density of these batteries. It is expected that this trend will continue in the future, with ongoing research and development to introduce nano-materials and mixed-metal technology for higher energy density (Ritchie and Howard, 2006; Howard and Spotnitz, 2007). As this technology advances, fewer cells per pack and/or less material per cell may be achievable, which may reduce total material flows to the EV battery waste stream.

The mass and composition of the EV battery waste stream modeled here has considered only the cells within EV battery packs. For a typical Tesla Roadster EV battery, with 6800 cells weighing approximately 46 g each, the total cell weight is about 313 kg, but the entire battery mass is about 450 kg (Berdichevsky et al., 2006). The difference in weight is attributed to the battery pack casing, module components, electronic parts, thermal insulation, etc. (Dunn et al., 2012), which can account for anywhere between 10 and 30% of the EV battery weight. Considering this entire battery system, the actual material flow entering the waste stream would be even greater than estimates calculated here. However, including other EV battery pack components would require further modeling, as these components may have higher reuse potential than the batteries themselves and may not enter the waste stream at the same time as the LIB cells (Cready et al., 2003).

3.4.3. *Battery chemistry and form factors*

All results shown to this point have followed an assumed mixture of different cathode chemistries, and are based only on 18650 (cylindrical) form factors. In the future, composition of the EV battery waste stream will depend heavily on the actual cathode chemistries and form factors selected by auto manufacturers. For example, new EV models such as the GM Volt and Nissan Leaf employ prismatic cells, while the 18650 form factor continues to be used in the Tesla roadster BEVs. Though this MFA model enables a reasonable approximation of the material and economic flows of the EV battery waste stream, there is a need to assess the role that a single dominant chemistry and prismatic form factor could play in determining the volume, composition, economic value, and recycling potential of this waste stream. Using the baseline scenario estimation for EOL LIB packs, the number of cells and material mass were estimated for the following cases:

(A) Scenarios in which a single dominant lithium-ion cathode chemistry (18650 form-factor) would be employed in all EV batteries. The four candidate chemistries were Li_2CO_3 (LCO), $LiMn_2O_4$ (LMO), $LiFePO_4$ (LFP) and mixed metal (NCM), each modeled individually as a dominant cathode chemistry.
(B) A scenario in which prismatic cells instead of 18650 cells were used. The chemistry mix of EV battery waste stream was assumed to be same as in the 18650 scenario (10% LCO, 30% each of LMO, LFP and NCM cells), however here only the LCO cells were of 18650 form factor (consistent with adoption by some auto makers) while the remaining 90% of the LIBs consisted of prismatic cells. See SI. for scenario details.

When compared to the baseline EV LIB outflows (a mix of potential chemistries in 18650 cells), the number of lithium-ion cells in the waste stream was roughly the same for the different dominant chemistry scenarios. The LFP scenario resulted in the highest number of waste cells, approximately 35 billion, cumulatively between 2015 and 2040 (as compared to the baseline 30 billion cells). When considering prismatic cells, results showed an interesting dichotomy: the total number of cells in the EV battery waste stream would reduce drastically if most EVs employed prismatic cells to almost 4.4 billion cells in the waste stream between 2015 and 2040, but the net flow and type of materials into the waste stream would remain relatively constant. This consistency held across all different scenarios of cathode chemistry and form factor, which ranged between 1 and 1.5 million metric tons of battery waste on a cumulative basis between 2015 and 2040 (Fig. 8).

Further, the recyclability of the EV battery waste stream would also vary with the battery chemistry and form factor (Fig. 8). For instance, if the $LiMn_2O_4$ chemistry is predominantly used, then the EV battery waste stream would contain negligible amount of currently recyclable materials and large quantities of low value materials that are not recycled. However, its recycling can generate value if the currently non-recycled high value materials like lithium and manganese can be recovered (Fig. 9). Similarly, even though both the form factor scenarios (Chemistry mix "Base Case" and 90% prismatic) consisted of the same distribution of LIBs belonging to the four cathode chemistries in the waste stream, the fraction of recyclable materials is slightly higher in the case of cylindrical cells, which require more metallic casing components (typically aluminum or steel).

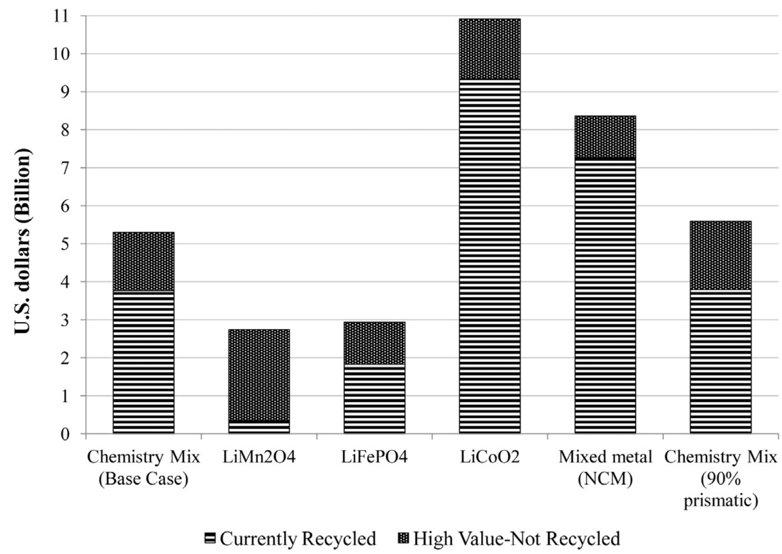

Fig. 9. Cumulative material value of EV battery waste stream (2015–2040).

An important caveat to these findings is that the high percentage by weight of recycled materials in the EV battery waste stream may not translate into high economic gains from LIB recycling. For instance, even though 58% of a $LiFePO_4$ cell waste stream consists of currently recycled materials, the relative economic value of this stream is lower than any other chemistry (Fig. 9). While the relative volumes of recycled materials are higher in the case of all cylindrical cells, as compared to the prismatic form, there is no significant difference in the total commodity value of materials in the waste stream associated with these two scenarios.

In fact, the economic feasibility of EV lithium-ion battery recycling in the future would not only depend on collection and recycling efficiencies, but also on the chemistries selected for EV battery manufacturing and ultimately ending up in the waste stream. A large scale use of LCO and NCM chemistries for EV batteries would translate into high economic values of the EV battery waste. In the LCO and NCM chemistry scenarios, currently recycled materials would constitute about 50% of the materials in the battery waste stream by mass, but could account for 86% of the economic value of that stream. However, as battery manufacturers shift to cheaper chemistries such as LMO and LFP, to improve performance and avoid high cost and scarcity of cobalt resources (Nishi, 2001), the resulting value of the currently recycled portion of the waste stream could be reduced to as low as 340 million USD, on a cumulative basis (2015–2040).

Cost efficient recycling procedures for the recovery of high value materials, like lithium and manganese, which are currently not recycled in the U.S. would add some incentive towards recycling of economically unattractive LIB chemistries. In the LMO chemistry scenario, currently non-recycled high value materials constitute 22% of the waste flows by mass, but account for 87% by value. Even more extreme, in the LFP battery chemistry scenario, the currently non-recycled high value materials accounted for only 1% of the waste EV battery cells by mass, but could make up to 38% of the total material value. Although Wang et al. (2014) conclude that for a LIB recycling facility to be profitable, the proportion of LiCoO2 cathode batteries in the waste stream needs to be 21%, improved recycling processes in the future is expected to improve the overall profitability of recycling EOL LIBs.

4. Conclusions and implications

It is clear that EV batteries will emerge as a future waste management challenge, with projected annual waste flows reaching as high as 340,000 metric tons by 2040. Because of the high volume, complexity and variety of materials forecast in the EV battery waste stream, it is evident that multiple waste management routes must be developed for EOL LIBs from electric vehicles:

(1) reuse avenues for battery cells and packs with remaining life,
(2) recycling infrastructure capable of recovering high value material from multiple battery chemistries, and
(3) safe disposal routes for materials with minimal or no secondary value or recovery infrastructure.

Results also indicate that high variability in the potential economic value associated with the projected LIB waste stream may pose challenges for development of recycling infrastructure. At present, profit from LIB recycling is constrained by high collection and processing costs (Wang et al., 2014). Currently, there is no federal regulation that mandates LIB recycling, and only two states – California and New York – have passed regulations banning landfill of these batteries. To overcome potential economic constraints of LIB recycling, particularly for less valuable, non-cobalt chemistries, the recovery process and infrastructure may require policy intervention to reach economies of scale.

Apart from economics, environmental health and safety may also motivate policy attention to future EOL LIB management. The absence of consistent infrastructure and regulations for lithium-ion battery recycling may increase the potential risk of environmental impact due to EOL EV batteries. Though the state of California classifies them as hazardous due to the presence of cobalt, LIB wastes are included under EPA's Universal Waste Rule (Gaines and Cuenca, 2000) and are in general not considered to be hazardous for the environment due to absence of toxic elements like lead, mercury or cadmium. However, landfill of EV LIBs may introduce environmental risks due to leakage of organic electrolytes, presence of heavy metals such as copper and nickel (Shin et al., 2005), reactive lithium salts, and large quantity of carbonaceous materials (graphite and carbon black).

A number of uncertainties still exist, and exact estimation of future waste flows will depend on the ability to further refine the forecasts of EV sales, battery and EV lifespan, and trajectories of battery technology deployment. Waiting until such refinements are possible, though, presents a risk of not allowing sufficient time for domestic infrastructure and policies to react to the emergence of a full scale battery waste stream. Thus, proactive advancement of

a robust EOL battery reuse, recycling, and disposal system will be required to handle the variety and volume of materials expected. Moreover, the MFA model provided here can be adapted to extend the analysis of LIB wastes as more definitive data become available.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.resconrec.2013.11.008.

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